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Original Article

Initial Effects of Woody Biomass Removal and Intercropping of Switchgrass (*Panicum virgatum*) on Herpetofauna in Eastern North Carolina

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ABSTRACT Forests are potential sources for a wide range of alternative fuels, which could reduce dependency on fossil fuels and carbon emissions, but sustainability of producing biofuels from forests has not been well-studied. Therefore, we investigated effects of woody biomass harvest, intercropping perennial grasses, and combinations of these treatments on herpetofauna in loblolly pine (*Pinus taeda*) plantations in a randomized and replicated field experiment in eastern North Carolina, USA. We sampled amphibians and small reptiles with drift fence arrays from April to July during 1 and 2 years after treatment establishment. We had 425 captures of 15 species of herpetofauna across the 2 sampling seasons, but did not observe large general effects of biomass removal or planting of switchgrass (*Panicum virgatum*) in pine plantations on detection, diversity, or relative abundance. However, planned contrasts indicated Simpson's index of diversity was greater in plots managed for switchgrass only compared with pine plantations during year 2, and that captures of southern toads (*Anaxyrus terrestris*) were less common in switchgrass plots than in pine plantations intercropped with switchgrass. Neither intercropping switchgrass with pine nor removal of harvest residuals caused herpetofauna diversity or abundance of common species to differ from traditional plantation management during the first 2 years following treatment establishment. With the exception of switchgrass-only plots, which had lower herpetofauna species evenness, the potential practices we considered for biofuels production are unlikely to have short-term effects on herpetofauna relative to traditional pine management. Future research should monitor herpetofauna through succession and consider landscape-scale effects and other potential feedstock sources.

KEY WORDS amphibians, biofuels, forest management, herpetofauna, intensive forestry, *Panicum virgatum*, pine plantations, reptiles, switchgrass.

Biofuels production is projected to increase rapidly in response to government mandates and incentives for generating alternatives to fossil fuels. For example, the Renewable Fuels Standards mandates that 36 billion gallons of renewable fuels be blended into liquid transportation fuels in the United States by 2022 (U.S. Department of Energy 2011). Further, 37 states have standards or goals in place that specify a minimum percentage of electricity production that must come from renewable sources by certain dates (U.S. Department of Energy 2011). Biofuels may address concerns with climate change, increasing energy demands, and reliance on foreign sources of oil, but whether their production is sustainable is debated (Fargione et al. 2008, Searchinger et al. 2008, Dale et al. 2010). Two concerns with biofuels are whether 1) developing biofuels markets will elicit a shift in agricultural production from food to fuel (Searchinger et al. 2008, Abbasi and Abbasi 2010), and 2) land management activities for biofuels production will be compatible with maintaining native fauna (Bies 2006, Fletcher et al. 2011, Riffell et al. 2011). Biofuels feedstocks can take many forms, from gleaning of crop residues in agricultural settings to growing and harvesting fast-growing woody crops or annual grasses in managed forests, each which may elicit different responses from biodiversity.

Intensively managed forests in the southeastern United States may provide several sources of biofuels feedstocks in addition to producing traditional timber products from the same land base. Recent research has examined feasibility and sustainability of growing dedicated feedstocks, harvesting forest residuals, and intercropping biofuels feedstocks in forest plantations, but numerous questions remain (Benjamin et al. 2010; Dale et al. 2010; Dymond et al.

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2010; Riffell et al. 2011, 2012). Intercropping biofuels on land where timber is produced will circumvent competition for food-producing land. However, influence of these management regimes on wildlife populations and their habitat is poorly understood (Riffell et al. 2012). In the southeastern United States, intercropping of switchgrass (*Panicum virgatum*), a U.S. Department of Energy model bioenergy crop, between rows of pines in forest plantations is occurring on a research scale across thousands of hectares of commercial forestland and is expected to increase as conversion facilities and associated markets develop (Wright and Turhollow 2010, Riffell et al. 2012). With the exception of short-term impacts on rodent populations (Marshall et al. 2012), influence of intercropping switchgrass in pine plantations on wildlife populations has not been investigated experimentally. Thus, forest managers and policy makers are lacking critical information necessary to understand whether potential sources of biofuels feedstocks derived from forests influence biodiversity and wildlife habitat (Riffell et al. 2011).

To aid in science-based management decisions regarding sustainability and biodiversity, we examined effects of a range of biofuels production options on herpetofauna. As a taxonomic group, amphibians and reptiles are an appropriate suite of species to evaluate influence of forest-based biofuels production because herpetofauna are linked to forest-floor microclimate and habitat structure, may influence below-ground soil processes, transfer energy flow between aquatic and terrestrial systems, and can be extremely abundant (Burton and Likens 1975, Welsh and Droege 2001, Walton 2005, Gibbons et al. 2006, Homyack et al. 2011). Further, our understanding of effects of habitat alteration on herpetofauna is limited due to underrepresentation in the scientific literature (DeStefano 2002, Christoffel and Lepczyk 2012).

Harvesting residual woody material for biofuels feedstock after clearcutting may alter habitat conditions negatively for amphibian and reptile species that need abundant downed woody debris to meet life-history requirements (but see Owens et al. 2008, Davis et al. 2010). The potential effects of a dedicated perennial grass feedstock in pine plantations on amphibians and reptiles are unclear (Riffell et al. 2012), but pasture and other open cover types can have negative effects on population persistence and individual movements of herpetofauna (deMaynadier and Hunter 1999, Rothermel 2004, Rothermel and Semlitsch 2006).

Our study is one component of a long-term investigation of environmental effects of harvesting woody biomass and intercropping perennial grasses in southeastern pine plantations. Our objective was to compare initial effects of different biofuels management techniques on relative abundance and species diversity of herpetofauna. We predicted that the range of biofuels options, including harvesting woody biomass, intercropping switchgrass with pine, and growing switchgrass alone would negatively affect amphibians and reptiles compared with typical intensive forest management because of changes to habitat structure that would impair their movements and/or reduce foraging, reproductive, and thermoregulation opportunities. To our knowledge, this study is the first randomized and replicated experiment to quantify influence of biofuels production on herpetofauna.

STUDY AREA

The Lenoir 1 Intercropping Sustainability Study is a collaborative experimental research study with industry, university, and government partners and was established and maintained by Catchlight Energy LLC, a joint venture between Chevron and Weyerhaeuser Company (Leggett and Sucre 2012). The study was located in eastern North Carolina in Lenoir County, USA, in a region dominated by commercial forestland and agriculture. As is typical for the region, a series of linear drainage ditches, which improve hydrologic conditions for pine growth and survival in plantations, occurred parallel to one another through the study area (Fig. 1). Pine trees were established using standard Weyerhaeuser methods, including clearcut harvest of the existing stand followed by mechanical and chemical site preparation, planting, vegetation management, and fertilization.

METHODS

The overall objective of this long-term study is to examine effects of intercropping and/or biomass management on sustainability and site productivity in a loblolly pine (*Pinus taeda*) plantation. The previous stand consisted of 109 ha of loblolly pine planted in 1974 and clearcut harvested in 2008. The study is a complete randomized block design with 5 treatments replicated 4 times ($n = 20$) on approximately 0.8-ha treatment plots (Fig. 1). Plots were separated by 12–50 m. Although plot sizes were relatively small for a study of wildlife responses, financial and logistic constraints prevented a larger scale experiment, and increasing the scope of inference by conducting a well-replicated, randomized

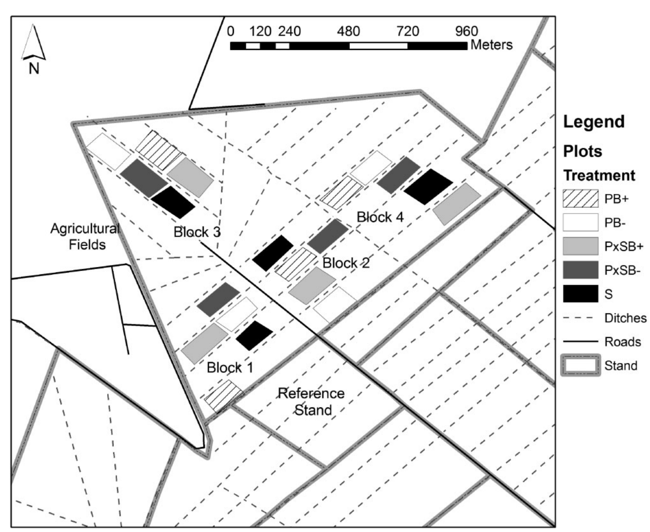

Figure 1. The effects of a range of biofuels feedstock treatments were examined across 4 experimental treatment blocks at the Lenoir 1 Sustainability Study site in eastern North Carolina, USA, 2010–2011. PB+ = pine with biomass in place; PB− = pine with biomass removed; P × SB+ = pine intercropped with switchgrass and biomass in place; P × SB− = pine intercropped with switchgrass and biomass removed; S = switchgrass only.

field experiment was an important benefit of the design (DeStefano 2002).

Treatments were installed with loblolly pine seedlings planted winter 2008 at approximately 1,100 trees/ha and switchgrass planted summer 2009 at 9 kg of pure live seed/ha using a modified corn planter. Treatments containing switchgrass incurred additional site preparation for the 3-m strips between crop tree rows (intercropped) or the entire plot to plant switchgrass (Leggett and Sucre 2012). Switchgrass was cut with a rotary mower after the first growing season in March 2010 and harvested December 2010 with a mower, rake, and round-baler.

The 5 treatments were:

1. Traditional pine establishment with biomass left in place (non-merchantable material left on site; PB+). This treatment serves as a control because it represents normal operations.
2. Traditional pine establishment with biomass removed (material that could potentially be used for biofuel production was removed; PB−).
3. Intercrop pine–switchgrass with biomass left in place (PS × B+).
4. Intercrop pine–switchgrass establishment with biomass removed (PS × B−).
5. Switchgrass only (S).

Site preparation varied by treatment. Treatments with pine were V-sheared and bedded using a bulldozer with required attachments to create a raised planting surface for pines. Beds had liquid suspension-based fertilizer incorporated into them to promote seedling root development. For biomass-removed treatments (B−), an excavator removed residual woody debris after clearcut harvesting to simulate a biofuels removal. An average of 9.4 Mg/ha of biomass remained on B+ treatments and 1.5 Mg/ha of biomass remained on B− treatments (Beauvais 2010). Intercropped switchgrass treatments (P × SB+, P × SB−) incurred additional V-shearing to prepare a 3-m strip between crop tree rows for planting switchgrass. For the switchgrass-only (S) plots, the entire plot was V-sheared and root-raked. We used Weyerhaeuser's coated Arborite® (Weyerhaeuser Company, Federal Way, WA) fertilizer to fertilize switchgrass plots in 2009 and 2, 4-D and a post-emergent herbicide (Basagran; BASF Corporation, Research Triangle Park, NC) to spot-treat vegetation competing with switchgrass during summer 2010.

Vegetation structure varied by treatment, and by the first year of the study, switchgrass and pine trees were well-established in treatment plots. During autumn 2010, mean cover of native and planted grasses was >95% on S plots, 66–75% on intercropped plots, and 49–65% on pine-only plots (Marshall 2011, Marshall et al. 2012). Woody debris cover was absent (0%) from S plots, covered 3–4% of biomass-removed plots, and covered 13–16% of biomass-in-place plots. Pines covered 5–14% of plots with pine and were absent in S plots. Vertical structure in pine and intercropped treatments was provided by pines (1.1–1.8 m ht), grasses (0.9–1.1 m ht) and forbs (1.0–1.2 m ht). In S plots, grasses averaged 1.5 m and forbs were 0.3 m tall. Additional details about vegetation structure of treatment plots are provided by Marshall (2011) and Marshall et al. (2012).

Amphibian and Reptile Sampling

During January–April 2010, we established a drift fence array near the plot center (>30 m from plot edges) to sample amphibians and reptiles in each of the 20 research plots (Willson and Gibbons 2009). Each "Y-shaped" array was oriented following a random bearing and consisted of 3, 10-m arms constructed of 60-cm-tall silt fence with the bottom 15 cm buried into the soil. We buried 9.5-L buckets with drainage holes to serve as pitfall traps, with 2 buckets on each side of the array ends and 3 at the center intersection ($n = 9$/array), and we added a wet sponge for a moisture source and a piece of woody debris across the top of the bucket for shade. We installed pitfall traps to passively capture moving herpetofauna that intersected the array. Further, we vertically inserted a 2.5-m polyvinyl chloride tube in the ground at approximately 1 m from the end of each drift fence "arms" ($n = 3$) to sample hylid frogs, which use these tubes as refuges (Willson and Gibbons 2009).

For 3 days/week from 15 April to 17 July 2010 and from 11 April to 14 July 2011 (12 and 14 sampling periods, respectively), we removed lids from pitfalls and checked arrays for captures approximately every 24 h. During 2 sampling periods in 2010, excessive heat caused us to close arrays a day early. All plots were open during the same time periods, thus eliminating potential confounding interactions between weather and treatment. We added water to sponges or removed water from buckets as needed. Occasionally, some pitfall traps were closed when excess water could not be eliminated during periods of high rainfall. We removed all captured herpetofauna from pitfalls, identified them to species, sexed them when possible, measured them (snout–vent length; cm), and released them on the opposite side of the drift fence or returned them to the polyvinyl chloride tube where they were captured. During both seasons we batch-marked individual anurans, salamanders, and lizards with Visible Implant Elastomer (Northwest Marine Technologies, Shaw Island, WA). Each treatment plot within a block was assigned a specific color of Visible Implant Elastomer so we could determine whether individuals were recaptured and whether they moved among treatment plots. We received all appropriate permits for this research (East Carolina University Institutional Animal Care and Use Committees D24; North Carolina Wildlife Resources Commission Scientific Collection Permit 10-SC00435).

Analyses

To account for imperfect detection of common species and to better understand how environmental conditions may have influenced our observations, we modeled detection rates and occupancy for species with >100 captures. We developed an *a priori* set of 6 models that incorporated potential effects of year, treatment, mean temperature, and/or total rainfall on detection (p) and occupancy (ψ). We conducted analyses with a multi-season, single-species occupancy model with Program Presence 4.3 (MacKenzie et al. 2003, Hines 2006) and used an information-theoretic approach using Akaike's Information Criteria (AIC) to evaluate parsimony of models. We considered each 3-day sampling period as the primary period and year as season and kept colonization (γ) constant. We quantified mean temperature and total rainfall for sampling periods (from 1200 hr on days pitfall traps were opened to 1200 hr on days arrays were closed), with an onsite weather station that recorded temperature with a Hobo U23 Pro v2 Temperature logger and rainfall with a Hobo Data logger Rain Gauge, model RG-3 (Onset Computer Corporation, Cape Cod, MA). Temperature and rainfall were used as environmental covariates in our model set. We used the detection and occupancy results to evaluate whether detection varied among treatments, and then proceeded to analyses of relative abundance and species diversity.

Within a sampling year, we quantified species richness (no. of herpetofauna species captured/plot) and diversity with the Shannon–Wiener species diversity index (H') and Simpson's index of diversity ($1 - D$; McCune and Grace 2002) by treatment plot. We calculated relative abundance (mean no. of captures) of herpetofauna per plot as number of captures in a year divided by 100 trap-nights adjusted for closed traps. For species with >100 total captures (a natural break in the data), we analyzed species-specific effects of biofuels treatments and adjusted them by detection rates <1, if necessary. We analyzed data on species richness, H', $1 - \mathrm{D}$, and relative abundance with repeated-measures Analysis of Variance (ANOVA) using PROC MIXED in SAS 9.2 (SAS Institute, Cary, NC). We considered study block a random effect, treatment as a fixed effect, and year as a repeated effect, and our model included both main effects and 2-way interactions involving the main effects. Residuals appeared to be normally distributed on the basis of visual inspection of quantile–quantile plots. We used the variance components covariance structure with homogenous variances for the random effects because this structure resulted in a lower AIC than a covariance structure with heterogeneous variances (Moser and Macchiavelli 2002). We used the autoregressive covariance structure for the repeated statement given that measurements in time were correlated.

Because the study was not fully factorial, we used planned contrasts to evaluate specific hypotheses related to effects of biomass removal and/or presence of switchgrass on diversity and abundance of herpetofauna to increase our statistical power (Rosenthal and Rosnow 1985, Day and Quinn 1989). Specifically, we examined 1) effects of adding switchgrass to pine plantations by comparing PB+ and PB− to P × SB+ and P × SB−; 2) effect of woody biomass removal from pine plantations by comparing PB+ and P × SB+ to PB− and P × SB−; 3) whether biomass removal and switchgrass planting in pine plantations had an additive influence on herpetofauna by comparing PB+ and P × SB− to PB− and P × SB+; 4) effect of planting pine in areas with switchgrass by comparing P × SB+ and P × SB− to S; 5) whether areas managed for pine only differed from areas managed for switchgrass only by comparing PB+ and PB− to S; and 6) influence of intercropping by comparing PB+, PB−, and S to P × SB+ and P × SB−. Each contrast was performed with responses pooled across years if we found no significant interaction between treatment and year in the overall ANOVA model. We performed contrasts separately for each year if there was a significant treatment × year interaction in the overall ANOVA model.

RESULTS

During summer 2010, we had 265 captures of 11 species across 34 nights of sampling (5,940 trap-nights); and during 2011, we had 160 captures of 13 species of herpetofauna across 42 nights of sampling (7,488 trap-nights). Captures included 149 southern toads (*Anaxyrus terrestris*), 129 eastern narrowmouth toads (*Gastrophryne carolinensis*), 52 Fowler's toads (*A. fowleri*), 31 pine woods treefrogs (*Hyla femoralis*), 23 oak toads (*A. quercicus*), 12 ground skinks (*Scincella lateralis*), 10 marbled salamanders (*Ambystoma opacum*), 8 southeastern five-lined skinks (*Eumeces inexpectatus*), 3 worm snakes (*Carphophis amoenus*), 2 green anoles (*Anolis carolinensis*), 2 slimy salamanders (*Plethodon chlorobryonis*), and 1 each of a squirrel treefrog (*H. squirella*), eastern hognose snake (*Heterodon platirhinos*), eastern kingsnake (*Lampropeltis getula*), and yellow-bellied slider (*Trachemys scripta scripta*). Recaptures included 12 (8 of pine woods tree frogs, 2 of southern toads, 2 of Fowler's toads) and 23 (20 of pine woods tree frogs, 3 of southern toads) incidences in 2010 and 2011, respectively. No recaptured animal had a Visible Implant Elastomer tag from a different treatment plot than where it was marked initially. We retained data from southern toads and eastern narrowmouth toads for additional analyses of detection and effects of biofuels production because capture success was great enough to warrant statistical analysis (i.e., >100 captures).

Mean temperatures ranged from 15.9° C to 28.8° C in 2010 and from 17.9° C to 28.1° C in 2011. Total rainfall across 2010 sampling periods was 70.4 mm and ranged from 0 mm to 40.2 mm/sampling periods. In 2011, total rainfall across sampling periods was 158.6 mm and ranged from 0 mm to 56.2 mm/sampling period. From our hypothesized model set for southern toads, a model incorporating a year effect on detection and constant occupancy had the lowest AIC value and 30% of the model weight (Table 1). Estimates of detection and occupancy of southern toads for the most parsimonious model ($\Delta\mathrm{AIC} = 0$) ranged from 0.21 to 0.28 ± 0.03 standard error and 0.90 ± 0.07 standard error, respectively. For eastern narrowmouth toads, temperature was the most important covariate predicting detection, and occupancy was best predicted by a constant ($W_i = 0.85$). For the most parsimonious model ($\Delta\mathrm{AIC} = 0$), estimates of detection and occupancy of eastern narrowmouth toads ranged from 0.02 to 0.37 and 0.96 ± 0.05 standard error, respectively. Models incorporating treatment into detection estimates were not in the top 4 models for either species ($\Delta\mathrm{AIC} > 6.00$; Table 1), which indicated that detection rates among treatments were comparable. Therefore, we did not adjust abundances for imperfect detection among treatments prior to ANOVA.

Table 1. Models of detection and occupancy including Akaike's Information Criteria (AIC), change in AIC (ΔAIC), model weight (W_i) and number of parameters (K) for occupancy (ψ) and detection (p) of southern toads and eastern narrowmouth toads sampled on 5 biofuels production treatments at the Lenoir 1 Sustainability Site, North Carolina, USA, 2010–2011.

Model	AIC	ΔAIC	W_i	K
Southern toads				
ψ(.), p(year)	451.88	0.00	0.30	4
ψ(.), p(temp)	451.91	0.03	0.29	4
ψ(.), p(rain)	452.54	0.66	0.21	4
ψ(.), p(.)	453.07	1.19	0.16	3
ψ(treatment), p(.)	457.13	5.25	0.02	7
ψ(.), p(treatment)	458.30	6.42	0.01	7
Eastern narrowmouth toads				
ψ(.), p(temp)	411.84	0.00	0.85	4
ψ(.), p(rain)	415.34	3.50	0.15	4
ψ(.), p(year)	436.06	24.22	0.00	4
ψ(.), p(.)	447.57	35.73	0.00	3
ψ(.), p(treatment)	451.94	40.10	0.00	7
ψ(treatment), p(.)	452.43	40.59	0.00	7

Intercropped plots had more captures of southern toads compared with switchgrass-only plots (contrast 4, $F_{1,12} = 4.96$, $P = 0.046$), but there was no difference between these treatment groups for eastern narrowmouth toads, total captures of herpetofauna, H', or species richness (contrast 4, $F_{1,12} \leq 3.00$, $P \geq 0.109$; Table 2). Abundance of southern toads, species richness, total captures of herpetofauna, and H' were not affected by intercropping switchgrass in pine plantations (contrast 1, $F_{1,12} \leq 1.42$, $P \geq 0.257$) or removal of woody debris from pine plantations (contrast 2, $F_{1,12} \leq 2.79$, $P \geq 0.121$) and these effects were additive (contrast 3, $F_{1,12} \leq 0.35$, $P \geq 0.566$; Table 2).

Relative abundance of southern toads, eastern narrowmouth toads, or herpetofauna, or species richness or H', did not differ in pine plantations compared with areas managed solely for switchgrass production (contrast 5, $F_{1,12} \leq 3.43$, $P \geq 0.089$; Table 2). Further, intercropping pine and switchgrass had no effect on relative abundance of southern toads, eastern narrowmouth toads, or herpetofauna present, or on richness or H' compared with plots managed for either pine or switchgrass alone (contrast 6, $F_{1,12} \leq 2.34$, $P \geq 0.152$; Table 2).

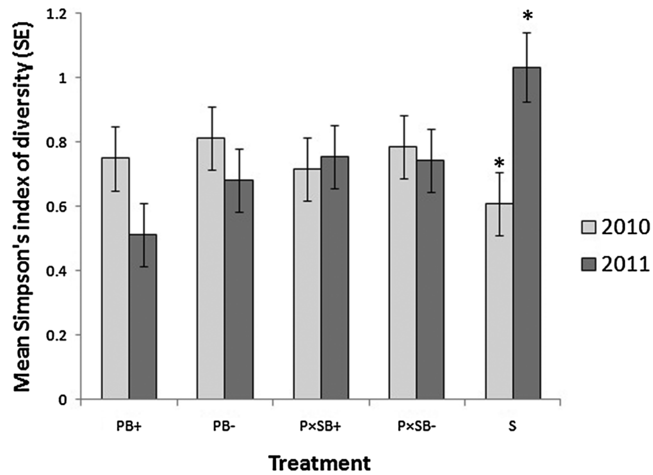

Figure 2. Simpson's index of diversity ($1 - D$) for herpetofauna sampled at the Lenoir 1 Sustainability Study site in eastern North Carolina, USA, during spring–summer 2010–2011. Amphibians and reptiles were sampled in areas managed for the production of biofuels for the first 2 years after treatment implementation. PB+ = pine with biomass in place; PB− = pine with biomass removed; P × SB+ = pine intercropped with switchgrass and biomass in place; P × SB− = pine intercropped with switchgrass and biomass removed; S = switchgrass only.

Abundance of all herpetofauna and abundance of eastern narrowmouth toads declined during the second year of study ($F_{1,12} \geq 11.82$, $P \leq 0.041$) to approximately half their values observed in year 1 (2010 $\bar{x} \pm 1$ SE: 4.28 ± 0.880 herpetofauna/100 trap-nights; 1.36 ± 0.311 eastern narrowmouth toads/100 trap-nights. 2011 $\bar{x} \pm 1$ SE: 2.24 ± 0.880 herpetofauna/100 trap-nights; 0.64 ± 0.311 eastern narrowmouth toads/100 trap-nights). We did not detect a significant change in abundance of southern toads, species richness, or Shannon's diversity index across years ($F_{1,12} \leq 9.25$, $P \geq 0.056$) or a treatment × year interaction ($F_{4,11} \leq 2.15$, $P \geq 0.137$).

In contrast, Simpson's diversity responded to treatments differently in each year (treatment × time, $F_{4,11} = 6.13$, $P = 0.008$; Fig. 2). Our planned contrasts revealed that neither addition of switchgrass to pine plantations, removal of woody biomass, or intercropping rather than managing areas for either pine or switchgrass alone affected Simpson's

Table 2. Least-squares mean and standard error estimates from the ANOVA model of amphibian and reptile richness, diversity, and abundance (mean captures/100 trap-nights) across treatments with a biomass removal harvest and/or planting of switchgrass (*Panicum virgatum*) in a loblolly pine (*Pinus taeda*) plantation in eastern North Carolina, USA, 2010–2011.

	Treatment[a]									
	PB+		PB−		P × SB+		P × SB−		S	
Variables	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Species richness	4.0	0.4	3.9	0.4	4.0	0.4	3.8	0.4	3.0	0.4
H'[b]	1.1	0.1	1.2	0.1	1.2	0.1	1.1	0.1	0.9	0.1
Total herptiles	4.2	1.1	2.7	1.1	4.1	1.1	3.1	1.1	2.1	1.1
Southern toad	1.5	0.5	0.9	0.5	1.5	0.5	1.3	0.5	0.6	0.5
Eastern narrowmouth toad	1.6	0.5	0.9	0.5	0.9	0.5	0.7	0.5	1.0	0.5

[a] PB+ = pine without a biomass harvest, PB− = pine with a biomass harvest, P × SB+ = pine intercropped with switchgrass and no biomass harvest, P × SB− = pine intercropped with switchgrass and a biomass harvest, S = switchgrass only with no residual biomass on site.
[b] Shannon–Wiener diversity index.

diversity in either year (contrasts 1, 2, 6, $F_{1,11} \leq 3.99$, $P \geq 0.071$). Further, intercropping switchgrass in pine plantations did not alter effects of woody biomass removal on Simpson's diversity in either year (contrast 3, $F_{1,11} \leq 1.42$, $P \geq 0.259$). However, Simpson's diversity was reduced when pine was planted with switchgrass relative to switchgrass-only plots in 2011 (contrast 4, $F_{1,11} = 7.51$, $P = 0.019$), but not in 2010 (contrast 4, $F_{1,11} = 2.36$, $P = 0.153$). Lastly, Simpson's diversity was greater in areas managed for switchgrass alone than in areas managed for pine alone in 2011 (contrast 5, $F_{1,11} = 17.64$, $P = 0.002$) but similar in 2010 (contrast 5, $F_{1,11} = 3.46$, $P = 0.090$).

DISCUSSION

We expected that removal of large amounts of coarse woody debris via biomass harvest and inclusion of switchgrass would have broad and consistent negative effects on herpetofauna by altering habitat structure, removing potential nest sites and opportunities for thermoregulation, and possibly hindering movements. Contrary to our predictions, biofuels feedstock treatments did not exert detectable effects on detection rates or occupancy of common species captured, relative abundance of herpetofauna, or on 2 of 3 metrics of diversity during this 2-year experimental field study (Tables 1 and 2). We did observe, however, that planting switchgrass alone enhanced Simpson's index of diversity relative to pine plots with or without intercropping, but this did not occur until 2 years after treatments were established.

Herpetofauna did not respond significantly to removal of nearly 85% of coarse woody debris up to 2.5 years after establishment of pine plantations. Similarly, experimental reduction of nearly all downed woody debris in mature loblolly pine stands had few effects on amphibians and reptiles for up to 10 years after removal in South Carolina, USA (Owens et al. 2008, Davis et al. 2010, Riffell et al. 2011). In another experiment, removal of coarse woody debris from recent clearcut harvests had less of a negative effect on small snakes than did removal of overstory trees, but woody debris coverage was low (0.8–4.8%) across all treatments (including clearcuts with retained debris; Todd and Andrews 2008). The values from Todd and Andrews (2008) compared with percent cover in our biomass-removed plots (3–4%). Owens et al. (2008) and Davis et al. (2010) suggested that herpetofauna of the southeastern coastal plain are more adapted to persist without the large amounts of coarse woody debris that are apparently integral to sustaining amphibian and reptile populations elsewhere (deMaynadier and Hunter 1995, Butts and McComb 2000, Hicks and Pearson 2003), possibly because frequent, low-intensity fires and high humidity historically held coarse woody debris levels low (Van Lear 1996).

Apparently, sufficient amounts of coarse woody debris remained after a biomass harvest in pine plantations at our study site to maintain herpetofauna within 2 years of the removal. Alternatively, it is possible we may have lacked sufficient power to detect statistically significant responses to coarse woody debris removal even though observed differences may have been biologically meaningful (*sensu* Owens et al. 2008 with $n = 3$). For example, total captures of herpetofauna in plots where coarse woody debris was removed was 70% of values in plots without coarse woody debris removal (contrast 2, $F_{1,12} = 2.79$, $P = 0.12$).

The potential importance of woody material to herpetofauna could also explain why complete removal of coarse woody debris in switchgrass-only plots (0% visual ground cover per Marshall et al. 2012) had significantly fewer southern toads relative to areas with debris present. Woody material serves as refugia for herpetofauna and has positive effects on water balance, energetics, and survival of individuals across a range of amphibian taxa (Rittenhouse et al. 2008, Homyack et al. 2011). The delayed response of Simpson's index to diversity to treatments at this site and the known positive effects of coarse woody debris on maintaining body water and increasing survival of amphibians and reptiles, suggests that herpetofauna may respond in the future and merits further examination.

Our study not only assessed consequences of removing coarse woody debris, but also explored whether managing for switchgrass as a possible biofuels source affected herpetofauna. Influence of intercropping perennial grasses on amphibians and reptiles has not been examined previously, but prior research investigated influence of non-forested patches (e.g., pasture, powerlines) on movements and survival of amphibians, particularly adults and metamorphosed juveniles that were leaving aquatic reproductive habitat. Juvenile American toads (*Anaxyrus americanus*) and spotted salamanders (*Ambystoma maculatum*) emigrating from experimental pools in a pasture were unsuccessful at reaching forested cover types >50 m from ponds (Rothermel 2004). Also, juvenile wood frogs (*Lithobates sylvaticus*) selected against open grass and shrub cover types on a powerline in Maine, USA (deMaynadier and Hunter 1999). Apparently, at least for pond-breeding amphibians, non-forested grassy cover types present either a mechanical barrier to dispersal or have inhospitable microclimates that can decrease survival and movements (Rothermel 2004, Rittenhouse and Semlitsch 2006).

We found that intercropping switchgrass in pine plantations had no effects on relative abundance of herpetofauna or species richness because metrics were nearly identical in PB+ and PB− treatments compared with P × SB+ and P × SB− treatments (Table 1). If switchgrass was a strong barrier to movements or did not provide suitable habitat, we would have expected fewer individuals to intersect drift fences and be captured in pine plantations with switchgrass than in pine plantations without switchgrass. In our study, switchgrass reached heights >1.0 m during the growing season (Marshall et al. 2012); and from the perspective of ground-dwelling amphibians or reptiles, it may have provided suitable canopy and microclimate conditions, unlike shorter grasses typically found in grazed pastures. Consequently, the decision to intercrop pine and switchgrass rather than planting pine alone may have no impact on herpetofauna abundance and diversity in the short-term.

Despite the weak effects of intercropping switchgrass in pine plantations on herpetofauna diversity and abundance, our results suggest that managing areas for monocultures of switchgrass could cause herpetofauna diversity and abundance to differ from pine plantations (with or without intercropping). In the second year (2011), Simpson's index of diversity values were greater in switchgrass-only plots than in either pine alone or pine intercropped with switchgrass, which indicated that diversity of amphibians and reptiles was altered as switchgrass became fully established. This index accounts for both evenness and richness, so that either metric could explain this interaction; however, we found relatively minor differences in species richness among treatments (Table 1). Therefore, the greater Simpson's diversity value in switchgrass-only plots was likely caused by an increase in species evenness. In support, relative abundance of the most commonly captured species, southern toads, was significantly lower in switchgrass-only plots compared with heterogeneous intercropped treatments, and 39% lower compared with pine-only treatments. This pattern suggests that southern toads responded positively to habitat structure provided by pine plantations, both alone and when intercropped with switchgrass. In another study, abundance of southern toads declined in clearcuts as succession occurred (Todd et al. 2009), but whether switchgrass will delay this response is unknown.

Both relative abundance of eastern narrowmouth toads and all herpetofauna were lower in the second year of the study, which suggested that populations may have responded to successional development after establishment of biofuels crops and pines or inter-annual environmental variation. Larger differences in diversity or abundances may occur as pine trees reach canopy closure and gain characteristics of mid-successional forest over the next few growing seasons. Habitat structure changes rapidly in southern pine plantations (Lane et al. 2011); and in our study, differences in pine and switchgrass height across all treatments were obvious between years. We expect that a shift away from a toad-dominated community will occur with canopy closure because toads are habitat generalists and can be abundant in recent clearcuts (Lannoo 2005, Todd et al. 2009). Alternatively, herpetofauna can display large inter-annual fluctuations (Beebee and Griffiths 2005), and may respond to environmental factors, but rainfall was not a primary factor affecting detection for common species in this study. However, we did not collect pre-treatment estimates or abundances from other areas during our study period, so we cannot determine definitively whether this temporal trend occurred in response to habitat succession or whether it reflects a larger scale trend in population size not attributable to our experimental manipulations. Finally, herpetofauna may respond to landscape-scale factors (Kolozsvary and Swihart 1999, Guerry and Hunter 2002), so that the surrounding matrix of commercial forestland and agriculture could have had a greater influence on abundances than did the smaller scale of our experimental treatments.

Currently, little information exists regarding influence of harvesting residual woody biomass, intercropping switchgrass, or combination of treatments on wildlife communities. Thus, our study, which examined a suite of biofuels production treatments on amphibians and small reptiles in a replicated and randomized experimental manipulation of southern pine forest, is valuable for understanding initial effects on one aspect of sustainability, herpetofauna biodiversity. Across the 2-year research project, neither gleaning of residual woody debris nor intercropping switchgrass had broad and consistent negative effects on herpetofauna when compared with traditional intensive forest management. Managing land for the sole production of switchgrass, however, may cause herpetofauna diversity and abundance to differ from areas managed for pine production for $\geq$2 years after establishment.

Future research should examine longer term responses of herpetofauna across feedstock production systems because habitat effects may be altered as planted pines approach canopy closure and switchgrass heights and densities increase or remaining coarse woody debris decays further. Intercropping switchgrass or other biofuels feedstocks (e.g., *Miscanthus* or short-rotation woody crops) in pine plantations would likely require wider row spacing (e.g., 6.1 m) and lower stocking densities than are typically used by many landowners, and these activities would likely increase habitat heterogeneity within forest stands with positive effects on species that prefer open-canopy forest. In addition, the small scale of research plots and trapping methods used in this study limited our sampling effort and excluded some reptiles with larger home ranges (e.g., large-bodied snakes) from capture so that response of herpetofauna to biofuel feedstocks should also occur on production-scale tracts of land with additional sampling devices, such as funnel traps and coverboards. Finally, the landscape-scale effects of biofuels production on population persistence of herpetofauna and other wildlife are generally unstudied and warrant additional examination (Riffell et al. 2012).

MANAGEMENT IMPLICATIONS

Based on the lack of large observed effects from intercropping switchgrass with pine trees or harvesting of residual biomass on herpetofauna diversity or abundance, these biofuels production regimes should be considered further as a potential source of feedstocks from forested systems. Plots planted with only switchgrass, however, may have greater herpetofauna diversity, which is driven by reducing reduction in the abundance of common species rather than by altering the number of species present.

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